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Maurice G. Estes Jr.
Research Scientist VI, maury.estes@nsstc.uah.edu

Mohammad Z. Al-Hamdan
University of Alabama - Huntsville

Jean T. Ellis
Universities Space Research Association, Huntsville, Alabama

Chaeli Judd
University of South Carolina - Columbia

Dana Woodruff
University of South Carolina - Columbia

See next page for additional authors

Authors
Maurice G. Estes Jr., Mohammad Z. Al-Hamdan, Jean T. Ellis, Chaeli Judd, Dana Woodruff, Ronald M. Thom, Dale Quattrochi, Brian Watson, Hugo Rodriguez, Hoyt Johnson III, and Tom Herder

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A MODELING SYSTEM TO ASSESS LAND COVER LAND USE CHANGE EFFECTS ON SAV HABITAT IN THE MOBILE BAY ESTUARY[1]

Maurice G. Estes, Jr., Mohammad Z. Al-Hamdan, Jean T. Ellis, Chaeli Judd, Dana Woodruff, Ronald M. Thom, Dale Quattrochi, Brian Watson, Hugo Rodriguez, Hoyt Johnson, III, and Tom Herder[2]

ABSTRACT: Estuarine ecosystems are largely influenced by watersheds directly connected to them. In the Mobile Bay, Alabama watersheds we examined the effect of land cover and land use (LCLU) changes on discharge rate, water properties, and submerged aquatic vegetation, including freshwater macrophytes and seagrasses, throughout the estuary. LCLU scenarios from 1948, 1992, 2001, and 2030 were used to influence watershed and hydrodynamic models and evaluate the impact of LCLU change on shallow aquatic ecosystems. Overall, our modeling results found that LCLU changes increased freshwater flows into Mobile Bay altering temperature, salinity, and total suspended sediments (TSS). Increased urban land uses coupled with decreased agricultural/pasture lands reduced TSS in the water column. However, increased urbanization or agricultural/pasture land coupled with decreased forest land resulted in higher TSS concentrations. Higher sediment loads were usually strongly correlated with higher TSS levels, except in areas where a large extent of wetlands retained sediment discharged during rainfall events. The modeling results indicated improved water clarity in the shallow aquatic regions of Mississippi Sound and degraded water clarity in the Wolf Bay estuary. This integrated modeling approach will provide new knowledge and tools for coastal resource managers to manage shallow aquatic habitats that provide critical ecosystem services.

(KEY TERMS: land use; hydrologic; aquatic; ecosystem; modeling; Mobile Bay; Gulf of Mexico.)

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INTRODUCTION

Estuarine ecosystems are influenced both by the watersheds that drain into them and by the ocean. The amount of freshwater runoff relative to the oceanic input of saltwater affects the salinity of the estuary. Furthermore, the turbidity generated by runoff, suspended sediments, and nutrient-driven phytoplankton production can limit the growth, distribution, and

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[2]Research Scientist VI (Estes), Earth System Science Center, University of Alabama in Huntsville, 320 Sparkman Dr., Huntsville, Alabama 35805; Senior Research Scientist (Al-Hamdan), Earth Science Department, Universities Space Research Association, Huntsville, Alabama 35805; Assistant Professor (Ellis), Department of Geography and Marine Science Program, University of South Carolina, Columbia, South Carolina 29208; Research Scientist (Judd), Senior Research Scientist (Woodruff), and Staff Scientist (Thom), Pacific Northwest National Laboratory, Sequim, Washington 98382; Senior Research Scientist (Quattrochi), Earth Science Department, NASA/MSFC, Huntsville, Alabama 35805; Director of Water Resources (Watson) and Water Resources Engineer (Rodriguez), Tetra Tech, Atlanta, Georgia 30339; Project Manager (Johnson), Terra Systems Southwest, Tucson, Arizona 86303; and Watershed Protection Coordinator (Herder), Mobile Bay National Estuary Program, Mobile, Alabama 36615 (E-Mail/Estes: maury.estes@nsstc.uah.edu).

abundance of submerged aquatic vegetation (SAV) as the range is limited to areas that receive sufficient photosynthetically active radiation (PAR). The presence and extent of seagrasses is employed as an indicator of water quality in some coastal areas (Boyer *et al.*, 1999; Hughes *et al.*, 2009). The SAV (including freshwater macrophytes and seagrasses) form extensive meadows in most temperate and subtropical estuaries, and are among the most valued habitats in the world because of their contributions to fisheries resource production (Costanza *et al.*, 1997). In Mississippi Sound, shoal grass and widgeon grass are commonly found, whereas the most extensive habitats in north Mobile Bay were the Eurasian watermilfoil, southern naiad, and wild celery (Vittor and Associates, 2009). Changes in watershed land cover caused by land use conversions can have dramatic effects on the runoff of sediments, nutrients, and potentially freshwater volumes (e.g., Hopkinson and Vallino, 1995; Thom *et al.*, 2001; D'Elia *et al.*, 2003). Orth *et al.* (2006) cite alteration of land cover in watersheds as one of the key stressors driving the global crisis for seagrass ecosystems. For this reason, restoration of estuaries and adjacent coastal ecosystems has focused on larger watersheds (e.g., Rabalais *et al.*, 2002; Valiela and Bowen, 2002; Scavia *et al.*, 2003; Mitsch and Day, 2006).

The historical SAV extent from the 1940s to 2009 has significantly declined (Figure 1). The recorded SAV area for Mobile Bay and its adjacent estuaries from historical distribution and aerial surveys was 1,301 ha for the combined 1940-1955 period, 577 ha in 2002, and 560 ha in 2009 (historical photo-interpretation from NCRS Mobile and Baldwin counties and Vittor and Associates, 2002, 2009). The overall change in SAV area from 1955 to 2009 was a decline of 57%. In 1940 and 1955, SAV existed in the shallow water aquatic ecosystems of Mississippi Sound, Mobile Bay, and Wolf Bay in the southeastern portion of the region. From the 1940s until 2002, SAV extent declined by 88% in eastern Mobile Bay and shoreline areas adjacent to Baldwin County and 55% in western Mobile Bay and shoreline areas adjacent to Mobile County and Dauphin Island (Vittor and Associates, 2002, 2009). Vittor and Associates (2004) suggested that "the prominent decline and apparently persistent disappearance in acreage since the 1940s and 1950s indicates that human activity has altered habitats capable of supporting SAV." Since 2002, an additional reduction of 1,300 acres of SAV has occurred in the northern portion of Mobile Bay and the Delta area. Several areas have seen small increases in SAV, including the eastern shore near D'Olive Bay and areas south of Mobile County in Mississippi Sound. The purpose of our study was to explore the role of land cover and land use (LCLU) change in watersheds adjacent to shallow water ecosystems in coastal Alabama influencing changes in discharge flows and associated changes in water properties and SAV throughout the estuary. Our two major research objectives were to (1) develop and

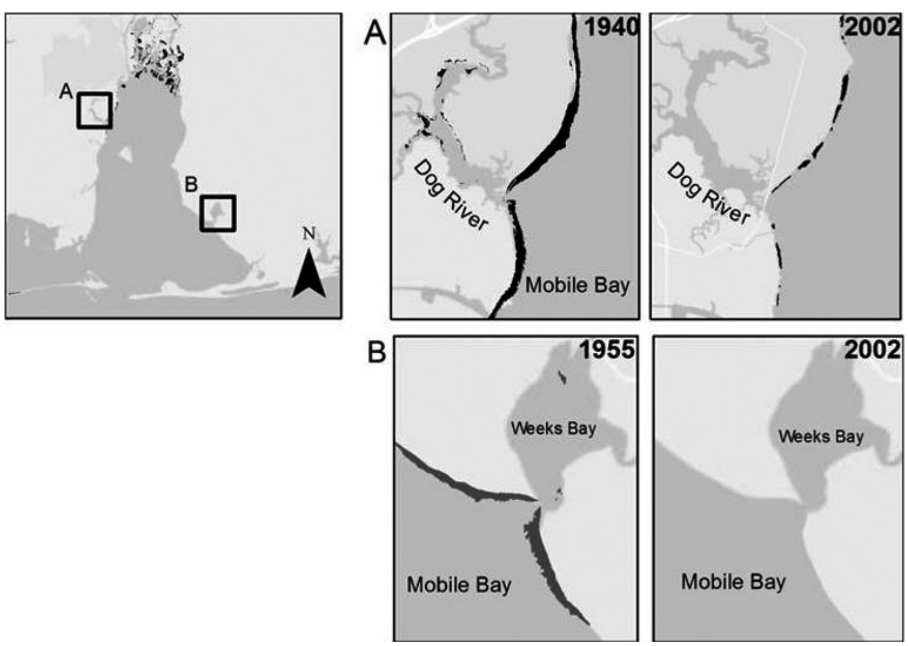

FIGURE 1. Historical Efforts to Map Mobile Bay Document a Loss in Areas Like Dog River (A) on the Western Shoreline and (B) Weeks Bay in Baldwin County. Submerged aquatic vegetation (SAV) distribution is displayed in black. In 2002, SAV was no longer found in Dog River and the distribution diminished along the northern and southern shoreline (A). Similarly, by 2002, SAV was not found in Weeks Bay (B).

utilize historical, present, and future LCLU scenarios for Mobile and Baldwin Counties, Alabama as input to models to predict the measured changes of water properties (temperature, salinity, and total suspended sediments [TSS]); and (2) predict the response of LCLU change on SAV distribution in the Bay and adjacent coastal areas. To investigate the potential linkage between changes in the watershed and changes in SAV, we considered alterations of water properties critical to SAV growth and distribution including temperature, salinity, and TSS. Temperature and salinity directly affect the SAV physiology, whereas TSS indirectly affects SAV through interference with subsurface light required for photosynthesis. The range of the seagrasses is limited to areas that receive sufficient PAR. PAR levels decrease with increases in turbidity — this is often strongly influenced by local hydrology, climate, and LCLU change (Koch, 2001; Thom *et al.*, 2008). Thus, changes in LCLU can decrease the extent of SAV (e.g., Dennison *et al.*, 1993).

Conservation and restoration of aquatic resources in Mobile Bay is a high priority for maintaining the ecological health of the system in this urban landscape. Stakeholder agencies have been focusing on local on-the-ground opportunities for conservation and restoration in the smaller, local subwatersheds. However, with a watershed that spans thousands of square miles, the benefit and impact of local actions on maintaining SAV meadows have not been assessed.

In the context of future LCLU changes, our study is intended to inform restoration and conservation actions in the watershed that would facilitate maintenance and expansion of Mobile Bay SAV meadows. We are evaluating the extent that LCLU changes driven by urbanization will significantly increase surface flows and impact salinity, water clarity, and temperature variables, which will affect SAV habitat suitability forecasts and thus impact future restoration and conservation efforts. In this study, we focused on changes in subwatersheds neighboring Mobile Bay and its adjacent estuaries, which are all located in Mobile and Baldwin Counties, Alabama.

The modeling approach used in this study is generally similar to that of other studies for other regions/watersheds (Hopkins *et al.*, 2000; Ciavola *et al.*, 2014). For example, Ciavola *et al.* (2014) studied how future changes in land use will affect runoff characteristics in 17 watersheds in Delaware-Maryland-Virginia Peninsula, the majority of which drains into Chesapeake Bay. The authors coupled demographic and urban growth models to create maps showing the location of predicted urban land use for the year 2030, and then used nutrient loading and rainfall-runoff models to forecast the consequences of such growth on the magnitude of changes that can occur in runoff quality and quantity.

Different modeling approaches that were used by other studies include using a tool developed by the U.S. Geological Survey (USGS) that can be helpful to the Bay restoration efforts, which is a set of spatially referenced regression models that relate suspended sediment and nutrient sources to stream loads. SPAtially Referenced Regressions on Watershed attributes (SPARROW) is the method used to build the regression models that retain and utilize detailed spatial information to statistically relate water quality measurements to suspended sediment and nutrient sources and the watershed characteristics that affect the transport of suspended sediments and nutrients throughout the watershed (Smith *et al.*, 1997; Preston *et al.*, 1998). Models of nutrients and suspended sediments using the SPARROW methodology were successfully developed and applied in the Chesapeake Bay watershed representing the late 1980s, early 1990s, late 1990s, and early 2000s (Brakebill and Preston, 1999, 2001, 2004; Preston and Brakebill, 1999; Brakebill *et al.*, 2001, 2010; Ator *et al.*, 2011).

STUDY SITE

Mobile Bay is a large estuarine system along the Gulf of Mexico (GOM) coast, lying within the state of Alabama in the United States (U.S.). Its mouth is formed by the Fort Morgan Peninsula on the eastern side and Dauphin Island, a barrier island on the western side. The Mobile and Tensaw Rivers empty into the northern end of the Bay, which creates the fourth largest streamflow discharge in the U.S. (Swann *et al.*, 2008) (Figure 2). The river delta system of Mobile Bay is the second largest in the U.S., which includes a network of wetlands and waterways that comprises over 200 rivers, bays, bayous, lakes, cutoffs, and sloughs. The Bay is 50 km (32 miles) long with a maximum width of 39 km (23 miles). The deepest areas of the Bay are located within the maintained shipping channel, sometimes in excess of 23 m (75 feet) deep, but the average depth is 3 m (10 feet) (Swann *et al.*, 2008). The unique characteristics of Mobile Bay resulted in the estuary being designated as an estuary of national significance in 1996 (http://www.mobilebaynep.com/land_use/).

A wide range of habitats and great species diversity are found in Mobile Bay. Habitat types include soft sediments, seagrass beds, barrier island dune and inter-dune wetland swales, fresh and saltwater marshes, pitcher plant bogs, bottomland hardwood forests, wet pine savannas, and upland pine-oak forests. There are three common SAV species in the

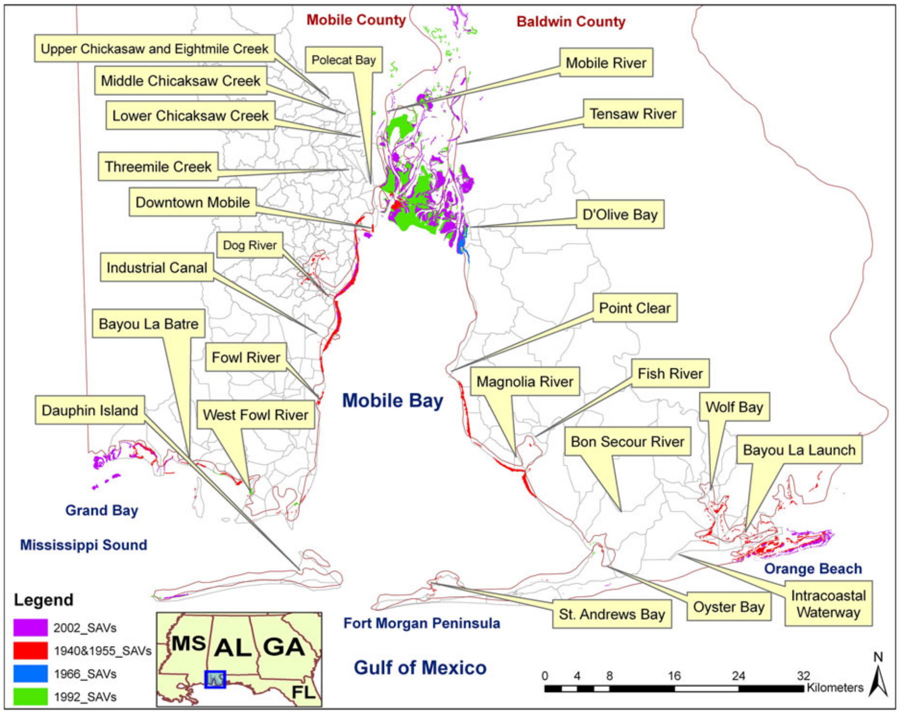

FIGURE 2. Location Map Showing the Counties, Discharge Points, and Historical Seagrass Coverage.

study area. In Mississippi sound, the seagrass species *Halodule wrightii* (shoalgrass) and *Ruppia maritima* (widgeon grass) are commonly found, whereas the most extensive habitats in north Mobile Bay include the freshwater macrophyte species *Vallisneria neotropicalis* along with other freshwater species (Vittor and Associates, 2009). Soft sediment habitats support fisheries resources critical to the local economy such as shrimp, oysters, and flounder. Approximately 98% of GOM harvested commercial fish and shellfish must spend a portion of their life span in estuaries and wetlands (NOAA-NMFS, 2009). Vegetated bottoms are one of the Gulf Coasts' most important ecosystems (Stout, 1998). SAV supports the estuarine food web through detrital and grazer pathways (Borum, 1979). SAV habitats also provide coverage for breeding and foraging important marine and estuarine species (Stout, 1998). Finally, SAV meadows provide physical structure that traps sediments. Because of its sensitivity to changes in water quality, SAV is sometimes used as an indicator of water quality in coastal areas (Boyer *et al.*, 1999; Hughes *et al.*, 2009).

The GOM wetlands comprise approximately 50% of all U.S. wetlands, which over the 1998-2004 period were being lost at a rate 25 times higher than the wetlands along the U.S. Atlantic coast (Envirocast, 2009; Gulf Restoration Network, 2009). These coastal wetlands provide critical storm attenuation capacity that protects human life and infrastructure, as well as crucial ecological resources that support fisheries, and multiple chemical and physical processes. This region's population is rapidly growing in some areas, which is driving the expansion of the Bay's working waterfront and port. From 2000 to 2006, the population growth of Mobile County was 1.1% and Baldwin Count 20.5% (U.S. Census Bureau, 2008). The remaining wetlands face an uncertain future with multiple anthropogenic and climatic sources of stress.

METHODOLOGY

Approach

An integrated modeling system comprised of watershed, hydrodynamic, and ecological models were developed for Mobile Bay to evaluate the impact of LCLU change in watersheds adjacent to the estuary on shallow aquatic ecosystems (Figure 3). The watershed model was used to evaluate watersheds contiguous to the Mobile Bay estuary (Figure 2). Watershed model output was linked with a hydrodynamic model to evaluate the impacts of LCLU change on Mobile Bay. Four modeling simulations were run with no differences except for the LCLU data for 1948, 1992, 2001, and 2030. All simulations were run using the same representative meteorological data that were collected at the Mobile Regional Airport

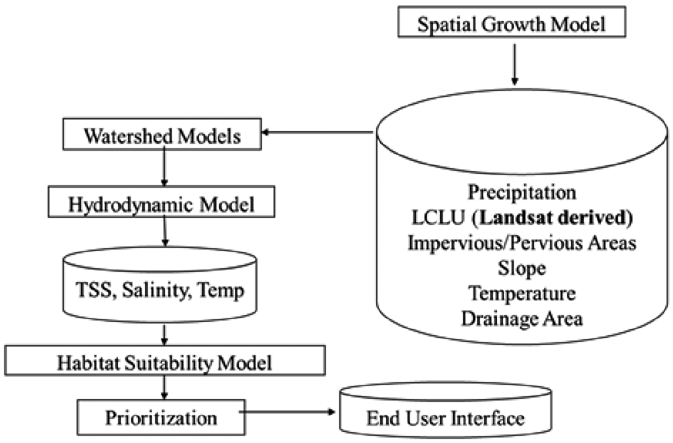

FIGURE 3. Flowchart Showing the Linkage between Modeling, Data Input, Prioritization, and User Interface.

and archived at the National Climatic Data Center (NCDC) from 2003 to 2005 (http://www.ncdc.noaa.gov/). All LCLU scenarios were developed to a common land classification system developed by merging the 1992 and 2001 National Land Cover Data (NLCD) (Table 1). 1992 and 2001 NLCD were used for Mobile and Baldwin Counties to determine recent historical trends and to serve as LCLU input data for spatial growth modeling and as inputs in the watershed model.

Watershed modeling was conducted to understand the impact of the projected urban development activities on the water quantity and quality discharging into the Mobile Bay estuary. The Loading Simulation Program in C++ (LSPC) model (USEPA) was used to simulate hydrology and sediments at the watershed scale. LCLU scenarios were developed for 1992 and 2001 from the NLCD and in 2030 using the Prescott Spatial Growth Model (PSGM) to evaluate impacts of flows into Mobile Bay from adjacent watersheds (Estes *et al.*, 2010). A map of 1948 land cover was used to represent "green" or low development conditions. All variables except LCLU were held constant for each model run. Impervious values for developed classes of low-density residential, medium- to high-density residential and urban commercial were developed from reference table values with adjustments made using the 2001 NLCD impervious surface area for developed classes (USDA, 1986; Yang *et al.*, 2003). A weighted average was computed for the developed classes between Mobile and Baldwin Counties to derive a normalized value for the study area (Table 2). Air temperature, precipitation, wind speed, and potential evapotranspiration from the Mobile Regional Airport for the period 2003-2005 were used as watershed and hydrodynamic models input.

The LSPC watershed model output provides changes in flow, temperature, sediments, and general water quality for the 22 discharge points into the Bay (Figure 2). These outputs were incorporated into the Environmental Fluid Dynamics Computer Code (EFDC) hydrodynamic model (Hamrick, 1992; Park *et al.*, 1995; Hamrick and Wu, 1997) to generate data on changes in temperature, salinity, and sediment concentrations on a ½ to 1-km curvilinear grid with four vertical profiles throughout the Bay's aquatic ecosystems. The models were calibrated using *in situ* water quality data collected at sampling stations in and around Mobile Bay in 2007. The watershed model inputs and outputs were on an hourly resolution. The hydrodynamic model inputs were on hourly resolution and its outputs were on bihourly resolution. Finally, light availability for SAV was mapped for each of the four modeling scenarios using a 30-m spatial grid, and evaluating, if modeled light levels met known SAV light requirements for each grid cell

TABLE 1. Remapping Land Cover Land Use Classes of 1992 and 2001 National Land Cover Data to a Common Classification.

Mobile and Baldwin Counties		
1992 Land Use Name	**2001 Land Use Name**	**New Class Name**
Water	Water	Water
Low-Intensity Residential, Urban Recreational Grasses	Developed Open Space, Developed Low Intensity	Urban Low-Density Residential/Recreational
High-Intensity Residential	High-Density Residential, Developed Med. Intensity	Urban Medium/High Density Residential
Comm/Ind/Transportation	Developed High Intensity	Urban Commercial
Bare Rock/Sand/Clay Quarries/ Strip Mines/Gravel Pits Transitional	Barren Land	Bare Soil/Transitional
Deciduous Forest	Deciduous Forest	Deciduous Forest
Evergreen Forest	Evergreen Forest	Evergreen Forest
Mixed Forest, Shrubland	Mixed Forest, Shrubs/Scrub	Mixed Forest/Shrub
Grassland/Herbaceous, Fallow, Orchards, Pasture/Hay, Row Crops	Grassland, Pasture Hay, Cultivated Crops	Agriculture/Pastures
Woody Wetlands	Woody Wetlands	Woody Wetlands
Emergent Herb. Wetlands	Emergent Herb. Wetlands	Emergent Herb. Wetlands

TABLE 2. Percent Impervious for Each Developed Land Cover and Land Use (LCLU) Class.

LCLU Class	Derived %	Table %
Urban Low-Density	5	12
Urban Medium/High-Density	23	36
Urban/Commercial	57	79

at each time step. The finer spatial scale of 30 m was used since this was the native scale of the bathymetry data from the National Geophysical Data Center, NOAA VDatum Digital Elevation Model Project (http://www.ngdc.noaa.gov/mgg/inundation/vdatum/vdatum.html). Predicted suitable SAV habitat extent was developed for each of the four modeling scenarios using a >19% surface irradiance (SI) threshold for habitat to be designated suitable.

Models Description and Calibration

Watershed hydrology plays an important role in the determination of nonpoint source flow and ultimately nonpoint source loadings to a water body. The watershed model must appropriately represent the spatial and temporal variability of hydrological characteristics within a watershed. Key hydrological characteristics include interception storage capacities, infiltration properties, evaporation and transpiration rates, and watershed slope and roughness.

LSPC Watershed Model. The LSPC is an extensive data management and modeling system capable of representing loading, both flow and water quality, from nonpoint and point sources and simulating instream processes. It simulates flow, sediment, metals, nutrients, pesticides, other conventional pollutants, temperature, and pH for pervious and impervious lands and water bodies. LSPC's algorithms are identical to those in the Hydrologic Simulation Program FORTRAN (HSPF). The LSPC/HSPF modules used to represent watershed hydrology for total maximum daily load development included PWATER (water budget simulation for pervious land units) and IWATER (water budget simulation for impervious land units). A detailed description of relevant hydrological algorithms is presented in the HSPF User's Manual (Bicknell *et al.*, 1996). LSPC was configured to simulate the watershed as a series of hydrologically connected subwatersheds. The subwatersheds were delineated using topography data that were collected from the USGS National Elevation Dataset. They were delineated based on the size and shape of watersheds and to match the location of flow and water quality monitoring stations, also shown in Figure 4.

LSPC Calibration. Initial parameter selection was based on previous modeling efforts in the coastal region of Mobile Bay and the entirety of the Mobile Bay watershed from northern Alabama to Mobile Bay (TetraTech, 2006) along with parameter recommendations in BASINS Technical Note 6 (USEPA, 2000). Typical minimum and maximum ranges for hydrologic soil groups and land uses were adjusted, until an acceptable agreement was achieved between simulated and observed streamflow. Parameters were not adjusted outside the possible minimum and maximum ranges defined by the BASINS Technical Note 6, including evapotranspiration, infiltration, upper and lower zone storage, groundwater storage, and losses to the deep groundwater system. Information on the watersheds' topography, geology, climate, land use, and anthropogenic influences were used to assist in parameter adjustment.

During the LSPC calibration, weather data from a number of stations in the watershed and surrounding coastal areas were collected and compared. Ultimately, rainfall and temperature data collected at the Mobile Regional Airport were found to more accurately represent conditions at the USGS stations in the watershed and were used in the modeling. Potential evapotranspiration, which is another important weather forcing parameter, was calculated using the Hamon method (Hamon, 1961).

The watershed model was utilized to estimate flows for the contiguous watersheds. Areas of the Mobile River Delta were not included in the watershed modeling effort because of the uncertainty associated with transport and exchange between the Mobile and Tensaw Rivers. The LSPC watershed model was calibrated to five USGS continuous streamflow stations (blue squares in Figure 4a; see color figure in online version). These stations are not tidally influenced and therefore can accurately measure freshwater flow. These stations capture flow from diverse LCLU activities, vegetation, and soils that make up the Southern Pine Plains and Hills (TetraTech, 2006). For these five stations based on 2002 climate data from the Mobile Regional Airport, the average correlation coefficient was 0.77, the average root mean square error (RMSE) was 2.75 m^3/s, and the mean absolute error (MAE) was 3.86 m^3/s. MAE ranged from 14.67 m^3/s at Threemile Creek to 0.3 m^3/s at Fowl River. The final calibration was found to adequately represent low flows and the rising and recessional limbs of storm events (Figures 4c and 4d).

Sediment data were not available during storms to perform a robust calibration of the LSPC model. Only one station had 12 months of sediment data for model calibration, which indicated mean annual variance of 45%. These data challenges and calibration results

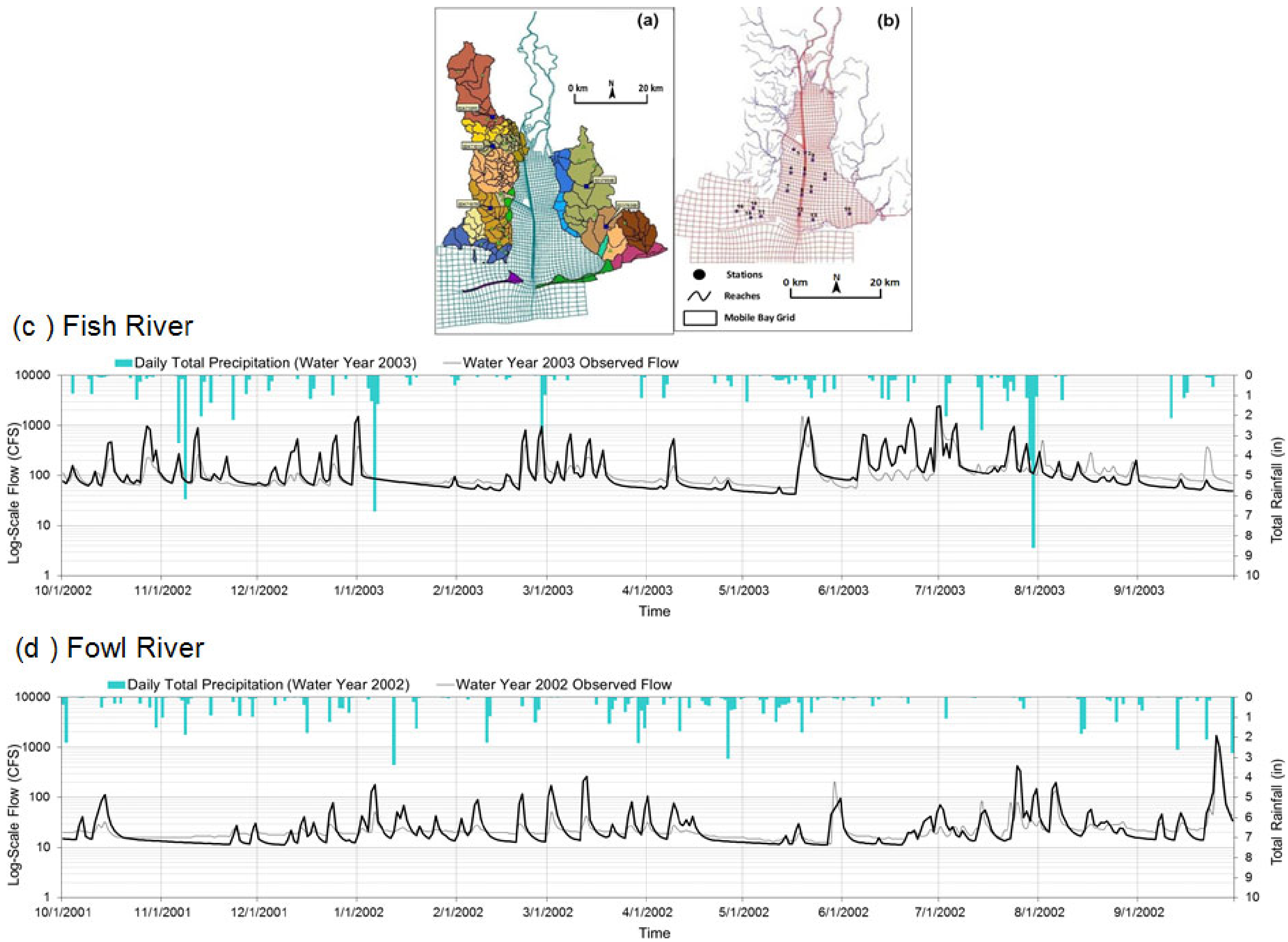

FIGURE 4. Locations of the Watersheds and Calibration Stations for the (a) Watershed Model and (b) Hydrodynamic Model (top) and Time-series Watershed Calibration for (c) Fish and (d) Fowl Rivers (bottom).

are comparable to other modeling studies (Donigian and Love, 2003; USEPA, 2006). Given data limitations and the amount of variance, this is the basis for the design of a sensitivity study to determine the effect of TSS variance on the modeling system (i.e., EFDC and habitat suitability modeling).

EFDC Hydrodynamic Model. The EFDC is a comprehensive three-dimensional model capable of simulating hydrodynamics, salinity, temperature, suspended sediment, water quality, and the fate of toxic materials. The EFDC hydrodynamic model takes into account wind speed and direction as well as tidal water surface elevation. The model does not account for wave dynamics; however, this would not be critical for this study in which we were studying the relative effects of LCLU change, and wave dynamics would have been similar among the different LCLU simulations. The model uses stretched or sigma vertical coordinates and Cartesian or curvilinear, orthogonal horizontal coordinates to represent the physical characteristics of a water body. The hydrodynamic portion of the model solves three-dimensional, vertically hydrostatic, free surface, turbulent averaged equations of motion for a variable-density fluid. Dynamically coupled transport equations for turbulent kinetic energy, turbulent length scale, salinity, and temperature are also solved. The EFDC model also simultaneously solves an arbitrary number of Eulerian transport-transformation equations for dissolved and suspended materials. The EFDC model allows for drying and wetting in shallow areas by a mass conservation scheme. The physics of the EFDC model and many aspects of the computational scheme are equivalent to the widely used Blumberg-Mellor model (Blumberg and Mellor, 1987) and U.S. Army Corps of Engineers' Chesapeake Bay model (Johnson *et al.*, 1993). A sediment transport model is part of the EFDC code (Hamrick, 2007). In general, sediment transport results from the flow of water over the surface of the sediment bed. The flow may be modified by the structure of the bed, debris within the flow field, and bends in the river. Erosion and deposition processes describe the link between the water column and bed sediment and their exchange rates. Solids are eroded and resuspended from the sediment bed due to the shear stress of moving water and solids in the water column are subject to advection and diffusion with the flowing water as well as settling due to gravity. The sediment transport submodel of EFDC simulates these interlinked processes mathematically.

The sediment transport module in EFDC solves the transport equation for suspended cohesive and noncohesive sediment for multiple size classes. Its capabilities include the following:

1. Simulates bed-load transport of multiple size classes of noncohesive sediment.
2. Simulates noncohesive and cohesive sediment settling, deposition, and resuspension/entrainment.
3. Uses a bed model that divides the bed into layers of varying thickness to represent vertical profiles in grain size distribution, porosity, bulk density, and fraction of sediment in each layer that is composed of specified size classes of cohesive and noncohesive sediment.
4. Simulates formation of an armored surficial layer.
5. Has a consolidation model to simulate consolidation of a bed composed of fine-grained sediment.

The EFDC model simulates the transport and fate in both the water column and sediment bed. Water column transport includes advection, diffusion, and settling. The sediment bed is represented by multiple layers with internal transport of contaminants by pore water advection and diffusion. Sediment and water are exchanged between the water column and bed by deposition and erosion.

As for the distinction between the different processes for cohesive and noncohesive sediment types; noncohesive, sand, sediment particles are generally larger in diameter and the particles are easily separable, whereas cohesive, silt and clay, sediment particles are small and adhere to each other as aggregates of hundreds or thousands of particles.

The application of the EFDC sediment transport model to Mobile Bay was applied in the same spatial domain used for the hydrodynamics (Figures 4a and 4b). It was applied with water surface elevation forcing at the GOM and Wolf Bay boundaries and freshwater inflows at the Mobile River just upstream of the Mobile-Tensaw split and various watersheds surrounding the bay.

The sediment boundary concentrations associated with Mobile River was estimated based on average sediment loads given in the literature. According to McKee and Baskaran (1999), the Alabama and Tombigbee Rivers entering the Mobile Bay system combine to an average annual sediment load of 4×10^9 kg/yr. Based on the annual average sediment load and annual average flow, an average annual suspended sediment concentration was calculated. The partition of the average suspended sediment load carried by Mobile River at the model boundary between cohesive and noncohesive sediment classes was initially estimated based on soil samples and adjusted during calibration.

The Mobile River discharge into the model was computed using USGS flows from the Tombigbee River at Coffeeville, Alabama, and from the Alabama River at Claiborne, Alabama. In accordance with the widely accepted approach of Schroeder (1978) to calculate the discharge of the system, the flows at these two gauging stations are added together and multiplied by 1.07.

As for the suspended sediment concentration at the open boundary, it was based on data collected from November 2007 to May 2008 in 16 stations in the GOM close to the main entrance to Mobile Bay. Based on these data, an average suspended sediment concentration value was estimated. The partition between cohesive and noncohesive was assumed the same as the Mobile Bay River boundary suspended sediment concentration.

Monthly averaged temperature data which were obtained by averaging temperature data collected at USGS stations (Tombigbee River below Coffeeville) and various stations on the Alabama River from 1991 to 1998 were applied to the upstream river flows. The water temperature measured at the Dauphin Island station was applied to the offshore open boundaries.

Water level data were obtained from the National Data Buoy Center, Dauphin Island, Alabama station. These data were used to generate the tidal boundary conditions. The boundary at the mouth of Mobile Bay is represented by tidal or water surface elevation. Water surface elevation data were not available as direct measurements in the GOM at the extent of our model boundary or from the east and west model boundaries in the Mississippi Sound and Perdido Bay for the modeled period. To generate the water surface elevation boundary forcing conditions, National Oceanic and Atmospheric Administration (NOAA) water surface elevation measurements at Dauphin Island, Alabama were utilized as the initial values for the south, west, and east boundaries. Values at the south offshore boundary were then calibrated by adjusting amplitudes and phasing to achieve the best comparison with measured data at Dauphin Island. For the east boundary at Perdido Bay, values were calibrated by adjusting amplitudes to achieve the net westward flow in the Gulf Intercoastal Waterway. Previous work documented this flow equal to ~1,000 cfs (Schroeder and Wiseman, 1986), but data collected by the Alabama Department of Environmental Management (ADEM) during neap and spring tides in 2007 measured average flows between 2,500 and 3,500 cfs.

EFDC Calibration. The calibration objectives for the hydrodynamic model were (1) to adequately represent the physics of the system by propagating momentum and energy based upon freshwater inflow, tides, and wind and (2) to model salinity intrusion and stratification because these factors play a major role in adequately representing water quality of the estuarine portion of Mobile Bay. Data were collected from 16 stations in Mobile Bay (Figure 4b) on a monthly basis between November 2007 and July 2008 as part of the Regional Sediment Management Program (Ellis and Kalcic, 2008) that was used in the model calibration. Additional information about the EFDC hydrodynamic calibration can be found in TetraTech (2006). Tidal water surface elevations, salinities, temperatures, flows, and TSS at various locations were used for calibration and quantitative assessments of the degree to which the model simulations match the observations were used to provide an evaluation of the model's predictive abilities (Figure 5). These assessments were made on the basis of different statistical characteristics of simulated and observed sets of data. For example, among the 14 stations throughout the Bay (Figure 4b), the TSS average Mean Error was 1.42 mg/l. The mean difference between the measured TSS concentration from field surveys and the

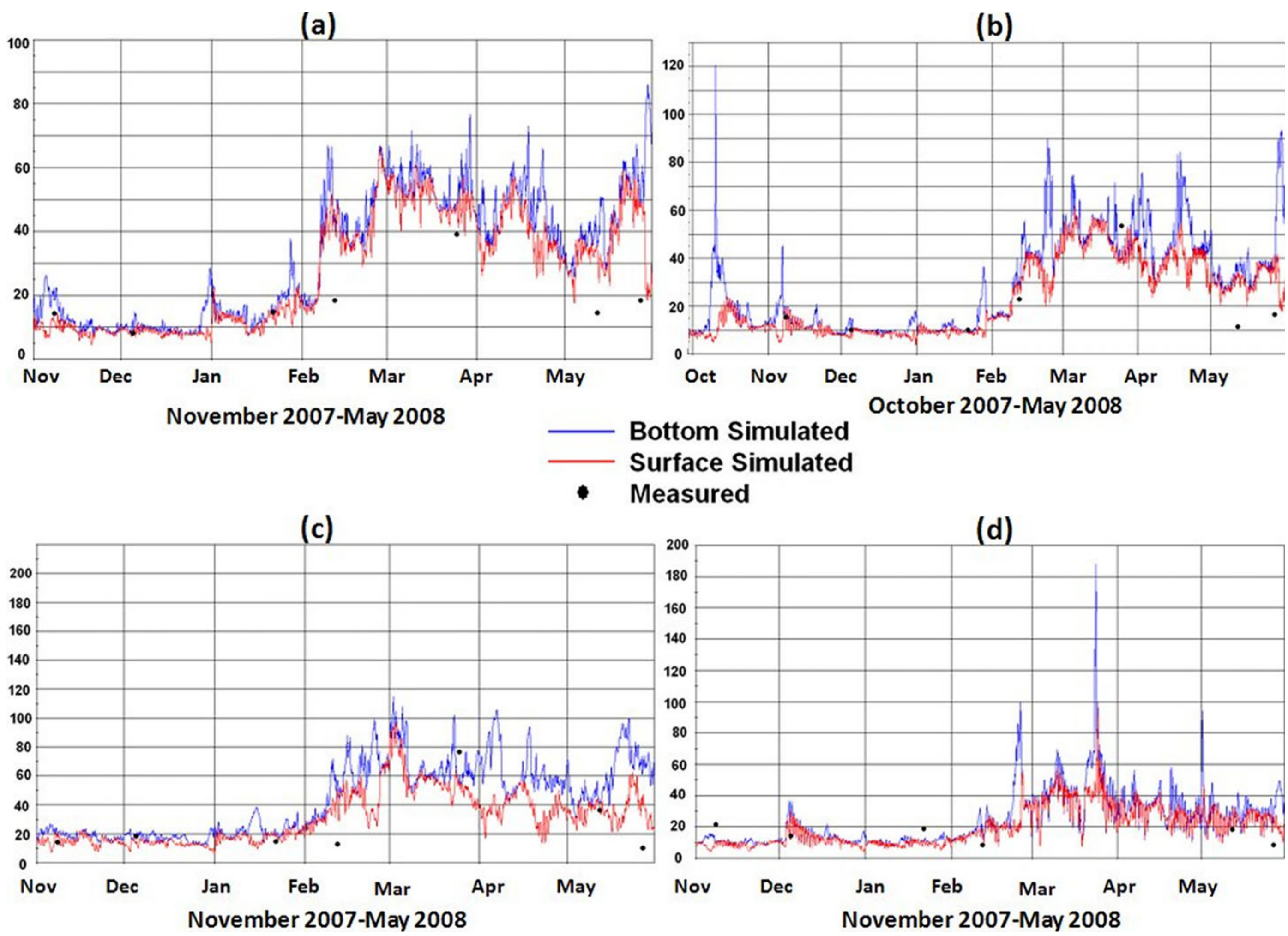

FIGURE 5. Hydrodynamic Model Total Suspended Sediments Calibration Plots for Bay Stations (a) 1 (North Bay), (b) 4 (West Middle Bay), (c) 6 (East Middle Bay), and (d) 13 (South Bay).

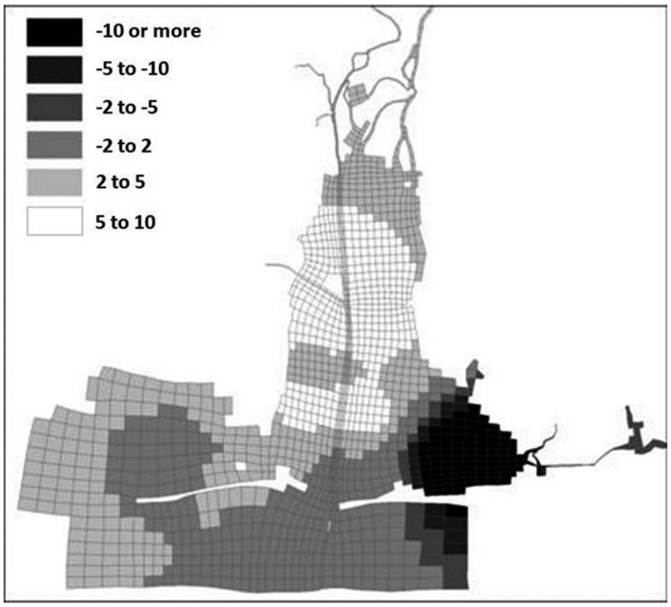

FIGURE 6. Total Suspended Sediments Mean Error (mg/l) in Modeled Surface Water Concentration Based on Comparison of Modeled Values with Field Sample Measurements.

predicted model concentration (mg/l) was recorded for each field survey point data point and was also extrapolated using an inverse distance weighted algorithm throughout Mobile Bay to yield a predicted average error for each grid cell as shown in Figure 6. In addition, the TSS average MAE was 5.83 mg/l, average RMSE was 13.12 mg/l, and average correlation coefficient was 0.4. The areas that had fewer stations (e.g., SE area) had, in general, higher errors in estimated TSS than those with more stations (e.g., SW area) (Figure 6).

The meteorological data needed for the hydrodynamic model calibration were obtained from the NOAA and the NCDC from 2003 to 2005. Meteorological parameters collected at the Mobile Regional airport station used in the modeling effort were air temperature, dew point temperature, cloud cover, precipitation, wind direction, and wind speed. Wind speed and direction data collected at the Dauphin Island station were accessed through NCDC and used for the southern portion of the model domain, while the wind speed and direction data from the Mobile Regional Airport station were applied to the northern portion. Other meteorological parameters were considered spatially uniform throughout the model domain.

During the model calibration, it was found that hurricanes Ivan and Katrina produced wind speeds up to 32 m/s which caused the model to become unstable. Therefore, to reduce instability, a speed of 10 m/s was used for these periods. Changes of wind speed and water surface elevation during the hurricanes are assumed not to affect the model results of other time periods that do not follow the hurricanes immediately.

NLCD Remapping

1992 and 2001 Landsat derived NLCD were used for Mobile and Baldwin Counties to determine recent historical trends and to serve as LCLU input data for spatial growth modeling and as inputs in the watershed model. However, these two products did not employ the same classification scheme, so it was necessary to remap the 1992 and 2001 NLCD classes to a common classification scheme for comparison between the two products (Table 1). LCLU data were used as input to the PSGM.

Wetland Normalization for 1948 LCLU

The 1948 LCLU map that was obtained from the state of Alabama included only four major classes (crop, crop/pasture, urban, and timber) (Figure 7), compared to the 10 detailed LCLU classes for 1992, 2001, and 2030 (Figure 7). Since the U.S. National Wetland Inventory program did not start until the 1970s, the 1948 LCLU map did not include a wetland class. To be consistent with the other LCLU scenarios, wetlands needed to be estimated in the 1948 LCLU map to input into the watershed model. We assumed that the wetlands distribution in 1948 was the same as that of the 1992 to fill this data gap. For the purpose of consistency, the 1948 major urban and forest classes were also broken down into more detailed subclasses as those of the rest of the LCLU scenarios. This was done by calculating the 1992 percentages of those detailed subclasses within the total areas of their major LCLU classes (i.e., urban and forests) and applied them to the 1948 total areas of those major classes. While this approach likely does not provide a 1948 wetlands class that is as accurate as the other 1948 classes, it was the best option with the data available.

Spatial Growth Modeling Method for 2030

The PSGM is a rule-based model that assigns future growth into available land based on user-defined parameters. Estes *et al.* (2010) provide additional details on the PSGM and its validation. U.S. Census Data were collected for Mobile and Baldwin Counties, including population for 1990 and 2000, and projections for 2005, 2010, 2015, 2020, and 2025 that are used to determine future land needs for resi-

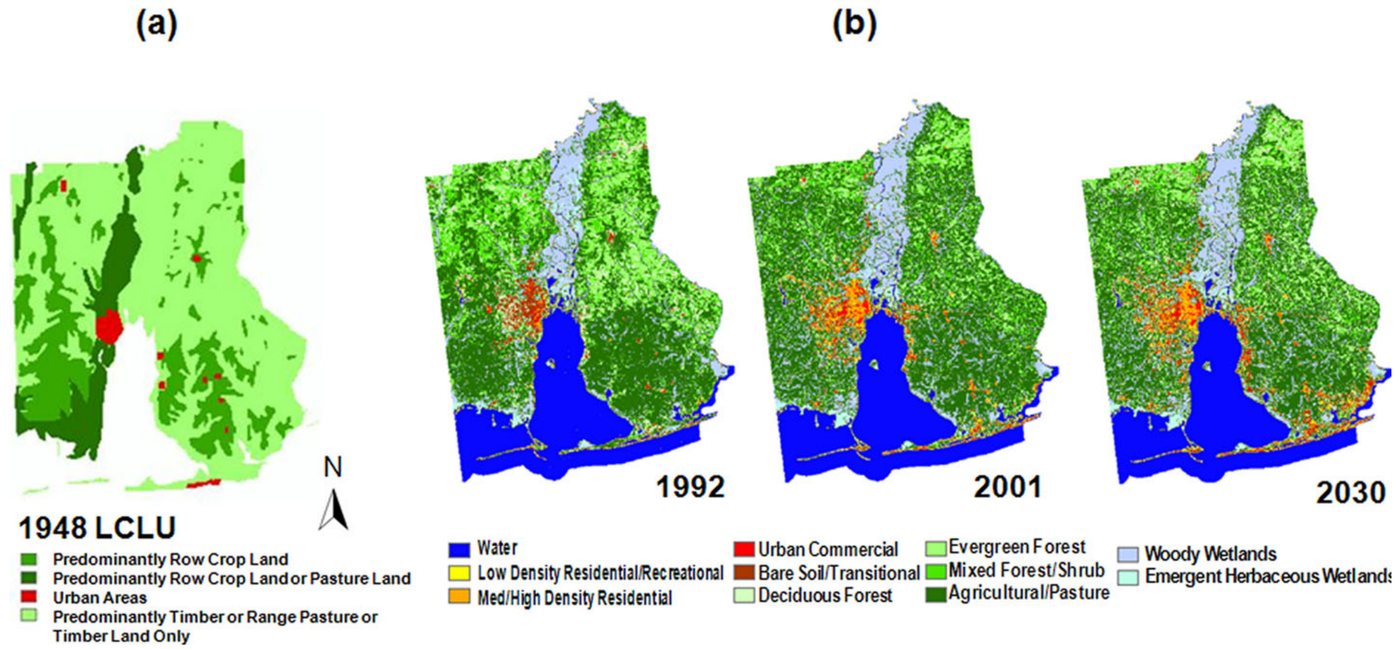

FIGURE 7. 1948 Historical Land Cover and Land Use (LCLU) Data for Input into the Watershed Model (left) and LCLU Data for Input into the Watershed Model for 1992, 2001, and 2030 (right).

dential development. Employment data for 1990 and 2000 were used to determine the number of jobs for each county, for each decade that is a driver for the amount of future commercial land needed. These data were used to determine trends, set model parameters, and define rule sets too. Additional model assumptions included growth continuing to follow existing LCLU trends, shoreline areas becoming growth attractors, and development being prohibited from wetland areas. Sufficient vacant land was available in 2030 to meet the projected demand for developed land use types within the study area, thus the rate of change projected by the model was not constrained.

Habitat Suitability Model

The objective of the habitat suitability analysis was to identify areas within Mobile Bay with a potential for expansion or reduction of SAV extent through time due to changes in water properties. A 30-m grid was used to cover Mobile Bay and the outlying coastal areas immediately adjacent to the mouth of the Bay. For each cell within the grid, and for each modeling scenario, a determination was made whether the cell would meet the light requirements for SAV or not, thus allowing change between scenarios to be evaluated on a cell by cell basis.

We used a five-step process to identify habitat suitability, using inputs from the EFDC model results. In the first two steps, we derived a statistical model to predict the water column PAR attenuation coefficient (K_{PAR}) from TSS concentrations, which were based on field measurements (Figure 8). This was necessary to translate modeled TSS into a biologically relevant light measurement for SAV. In the following three steps, we applied the statistical model in a geospatial assessment to identify areas of change. Finally, to validate the model, we compared areas of predicted high light availability in 2001 to field data for SAV occurrence (2002-2003). For the purpose of this analysis, we examined the changes during the month of May using averages of daily hydrodynamic model outputs. SAV is not highly stressed by temporary changes in turbidity of a few days or more, but a sustained decrease in light over time will impact survival (Moore *et al.*, 1997; Thom *et al.*, 2008). Spring is a critical time for SAV growth and carbohydrate storage, and May was selected as a representative month of this time period (Moore *et al.*, 1997).

The process we deployed to identify changes in predicted SAV habitat spatial distributions (Figure 8) was as follows.

1. Derive TSS-K_{PAR} Relationship. Acquire *in situ* TSS concentration and light attenuation data. The objective of this step was to acquire *in situ* measurements of light attenuation (K_{PAR}) concurrently with TSS concentration and to develop a statistical relationship between the two. Eight of the 16 stations from the Sediment Management Program (Figure 4b) were used for algorithm development because they

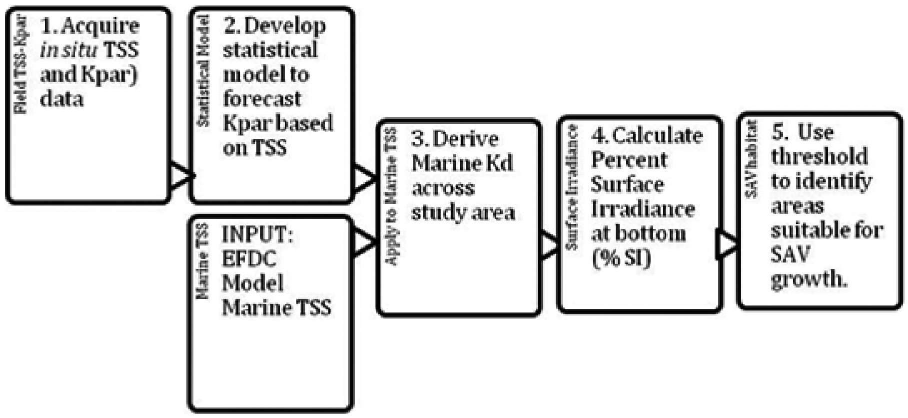

FIGURE 8. The Five Steps Used to Identify Changes in Predicted Submerged Aquatic Vegetation (SAV) Habitat in Mobile Bay.

provided a complete sampling record and the greatest range in sample concentrations.

TSS concentrations were determined gravimetrically based on dry-weight analysis from samples collected at 0.5-m depth. Concentrations ranged between 4.3 and 53.6 mg/l and were representative of the seasonal environmental conditions found in Mobile Bay during the periods sampled. The diffuse light attenuation coefficient for (K_{PAR}) is a fundamental measure of water clarity that is important for SAV growth and production in coastal areas. K_{PAR} is defined as the exponential rate of light decrease in the water column through Beers law:

$$K_{\mathrm{PAR}} = -\ln\left(\frac{E_z}{E_s}\right)/Z \tag{1}$$

where K_{PAR} is the diffuse attenuation coefficient expressed in units of reciprocal meters (m^{-1}), E_z is the irradiance at depth z, and E_s is the irradiance just below the water surface (Jerlov, 1976). K_{PAR} was determined from measurements of integrated irradiance (400-700 nm) taken at 0.5-1.0 m increments to mid-water depth using a LICOR LI-192 underwater quantum sensor (LI-COR Environmental Division, Lincoln, Nebraska). Using a modified form of Beers law, data taken at multiple depths per station were log-linear regressed against depth to determine K_{PAR}. Only regression coefficients values exceeding 0.975 were used for further algorithm development.

2. Develop Statistical Model to Predict K_{PAR} Based on TSS. To develop the relationship between K_{PAR} and TSS, several statistical models were explored to describe the data, including a linear, exponential, and logistic model. The logistic regression model (Equation 2, Figure 9) provided the best overall fit ($r^2 = 0.72$).

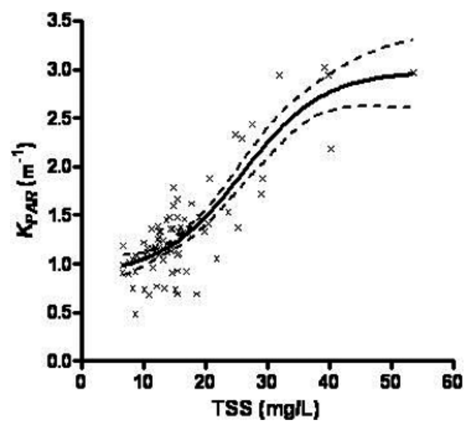

FIGURE 9. The Relationship between Total Suspended Sediments (TSS) and Field Measurements of K_{PAR}.

$$K_{\mathrm{PAR}} = 0.884 + \frac{2.104}{1 + 10^{(26.01-\mathrm{TSS})+0.665}} \tag{2}$$

In addition to TSS, other optically active components of water clarity (i.e., chlorophyll [chl] and dissolved organic carbon [DOC]) were statistically examined for their contribution to K_{PAR}. DOC data were collected in the field at the same time as TSS and K_{PAR}. While each of the components contributes to K_{PAR} (K_{PAR} = TSS + chl + DOC), TSS showed the greatest correlation with K_{PAR} ($r = 0.64$) and explained most of the variability in the relationship.

Predict Change in Habitat. Previous studies in Alabama indicate that the endemic seagrass *H. wrightii,* experiences light limitation at levels <19% SI (Shafer, 1999). Similar studies in Texas found limitation at 15-18% SI (Dunton, 1994). Assuming that the observed light limitation thresholds were similar among all SAV in the Bay, we used these thresholds to identify areas on a 30-m grid resolution that met the 19% SI threshold, based on the May monthly mean TSS values and bathymetry.

3. Derive Marine K_{PAR} across Study Area. Prior to deriving K_{PAR}, we calculated mean model error for TSS prediction by comparing *in situ* TSS concentrations from the sampling effort to modeled concentrations during the same time period. The EFDC-derived mean TSS concentration value for the month of May was recorded in each grid cell across the study area. The mean difference between the measured TSS concentration from field surveys and the predicted model concentration (mg/l) was recorded for each field survey point data point and extrapolated using an inverse distance weighted algorithm throughout Mobile Bay to yield a predicted average error for each grid cell. This approach assumed that the distance from predicted to observed point is the primary explanatory factor in variance rather than hydrological inputs, and that error seen in mean model data in May was similar to the mean seen throughout the year. While this is a generalization, it assists in identifying areas within the Bay where results should be viewed with some caution, such as Oyster Bay which showed a large TSS error (Figure 6).

We corrected EFDC TSS model values for the mean measured bias. Mean error was added to the TSS monthly values and used as inputs to the K_{PAR}-TSS algorithms developed in Step 2 to derive the K_{PAR} attenuation coefficient for each grid cell in the study area for 1948, 2001, and 2030. Outputs were converted into raster datasets with a 30-m resolution.

4. Calculate Mean Percent Surface Irradiance. In Equation (1), the term E_z/E_s represents the decimal percent SI at depth (z), reconfiguring the equation:

$$\frac{E_z}{E_s} = e^{(-K_{PAR} * Z)} \quad (3)$$

NOAA's 1/3 arc second (approx. 10-m resolution) VDATUM fused nearshore bathymetry dataset for Mobile, with a vertical datum of NAVD88, was imported into GIS (Amante *et al.*, 2010) and used to represent (z), the depth of SAV. In Mobile Bay, NAVD88 is approximately 0.07 m above MLLW. In ArcGIS 10.0, mean percent SI was calculated across the study area for each assessment year using the derived K_{PAR}. We also evaluated a secondary product that covered Wolf Bay, located east of Mobile Bay off the Intracoastal Waterway. However, the bathymetric values in this secondary dataset were recorded as integer values, and did not have the precision to capture changes of less than 1 m in depth. Thus, we could not evaluate Wolf Bay at this resolution; rather, we utilized the rapid assessment detailed below.

5. Use Threshold to Identify Areas Suitable for SAV Growth. Areas in Mobile Bay meeting the 19% SI threshold were identified for each time step (Kenworthy and Fonseca, 1996).

Rapid Assessment of Maximum Depth. In addition to the assessment above, a separate rapid assessment was carried out comparing seven sites, geographically capturing the extent of Mobile Bay: D'Olive Bay, Downtown Mobile, Fish River, Oyster Bay, Wolf Bay, Dog River, and Bayou LaBatre. Five to eight grid cells in these sites were used to identify areas where the maximum depth of SAV within the cell changed more than 0.1 m over the time series (1948-2030). Using the 19% threshold in Equation (3) (E_z/E_s term = 0.19) and the K_{PAR} values from each cell, we derived the maximum SAV depth for that cell for each scenario. When the change of maximum depth was greater than 0.1 m between scenarios, the cell was flagged.

Validation

The watershed model validation was performed by comparing modeled to observed flows in 2001 at four discharge stations spatially arranged around the Mobile Bay estuary; Fish River, Magnolia River, Chickasaw Creek, and Fowl River (Table 3). Overall, the mean errors range from 0.32 to 1.32 m^3/s and correlation coefficients from 0.43 to 0.77. The watershed model tends to underpredict maximum streamflows with the exception of Chickasaw Creek, which contributes to the lower correlation coefficients among the other three discharge sites evaluated.

Validation of the habitat suitability model was conducted by comparing areas that were predicted to receive ≥19% SI in 2001, which we considered to be areas of high light suitability, with locations where SAV was present in a 2002 SAV field survey (Vittor and Associates, 2002). We used the field survey to compare the number of sites surveyed with SAV present that fell in or near (<50 m) the high light suit-

TABLE 3. Validaton Results for the Observed *vs*. Modeled 2001 Streamflow Discharge.

	ME (cm)	MAE (cm)	RMSE (cm)	*R*
Fish River	0.32	1.19	1.51	0.57
Magnolia River	0.32	0.40	1.55	0.43
Chickasaw Creek	1.23	2.58	8.08	0.77
Fowl River	0.29	0.52	1.80	0.52

Note: MAE, mean absolute error; RMSE, root mean square error.

ability areas in our 2001 modeled scenario. In ArcGIS 10, we selected 27 field data points that fell within the model study domain where the freshwater SAV species *V. neotropicalis*, or the seagrass species *H. wrightii* or *R. maritima* were present, and identified points that were within 50 m of the high light suitability regions.

Of the 27 SAV field data points examined, 18 (66.7%) were within 50 m of projected high light suitability regions. Nine points were in projected low light areas, with a majority of these sites near the causeway. All of the sites with SAV outside Mobile Bay were in high light suitability areas.

Sensitivity Analyses

Simulations were performed based on a +50% and −50% change in TSS from the watershed model to determine the hydrodynamic and habitat suitability models sensitivity.

EFDC model simulations were developed by adding ±50% to the TSS boundary concentrations of the 2001 baseline simulation and the results compared. The sensitivity analysis results showed that the −50% simulation caused TSS changes within the Bay ranging from −30 to −60% with an average of −47.5% and a standard deviation of 4.9. Eighty-five percent of the 1,750 Bay grid cells had a change between −45 and −55% (Figure 10a). A majority (72%) of the grid cells had a change between −47.5 and −52.5%, which are the grid cells in the less dynamic regions of the Bay (Figure 11a). The sensitivity analysis results also showed that the +50% simulation caused TSS changes within the Bay ranging from +30 to +70% with an average of +48.5% and a standard deviation of 8.9. Seventy percent of the 1,750 Bay grid cells had a change between +45 and 55% (Figure 10b). A majority (54%) of the grid cells had a change between +47.5 and +52.5%, which are the grid cells in the less dynamic regions of the Bay (Figure 11b).

TABLE 4. Habitat Suitability Assessment for Baseline and Plus/Minus 50% Sediment. Differences between scenarios and baseline conditions range from 5 to 10% changes in suitable habitat.

Scenario	Area with Suitable Light (ha)
Baseline	99.6
Plus 50%	91.8
Minus 50%	114.8

For the habitat suitability model, we reevaluated light suitability comparing baseline conditions to those with output from the hydrodynamic model based on a 50% increase and decrease in suspended sediments. Results indicate a total change of suitable areas by 5-10% (Table 4). The most prominent areas of change were close to streams, while those areas farther from inputs, such as on the east side of Mobile Bay, showed little change (Figure 12).

The scenario of an increase in suspended sediments by 50% produced many TSS values that were outside the maximum values obtained from field data used to develop the original TSS/K_{PAR} model, so we used the maximum K_{PAR} value (K_{PAR} = 2.98) for these locations. This equated to an approximate maximum depth for SAV of −0.55 m. Thus, a 50% increase in TSS in an already very turbid area would predictably result in little change in suitable habitat area (Guisan and Zimmermann, 2000).

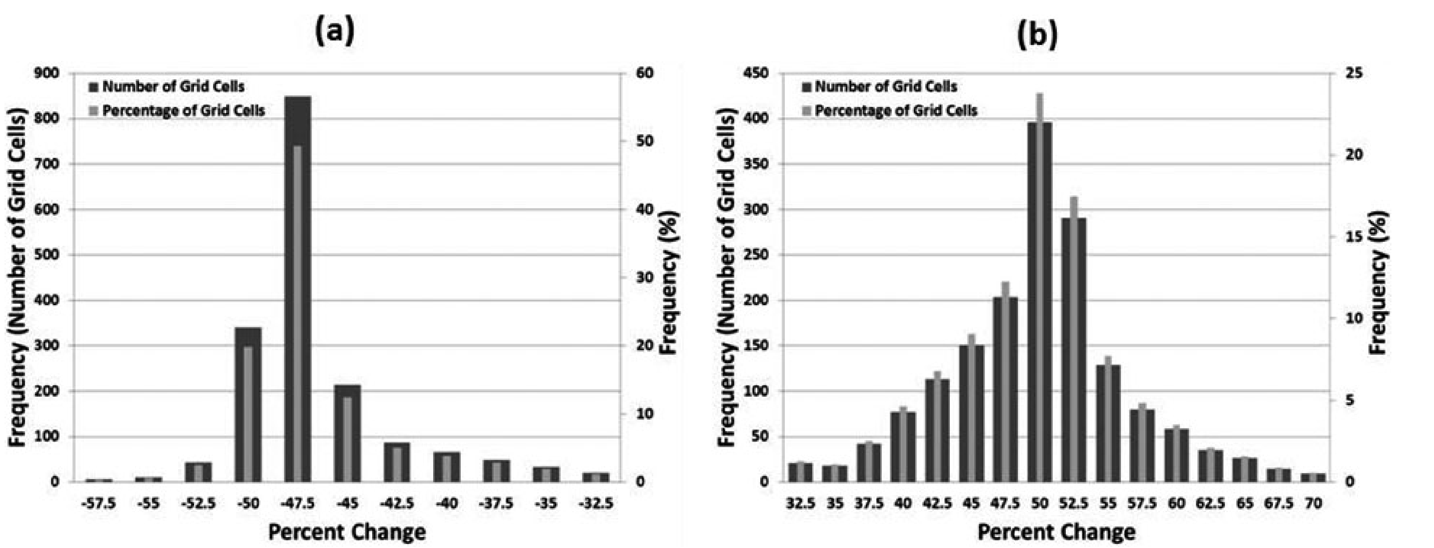

FIGURE 10. Histograms of the Total Suspended Sediments Changes within the Bay Grid Cells for the −50% Sensitivity Analysis Simulation (a) and the +50% Sensitivity Analysis Simulation (b).

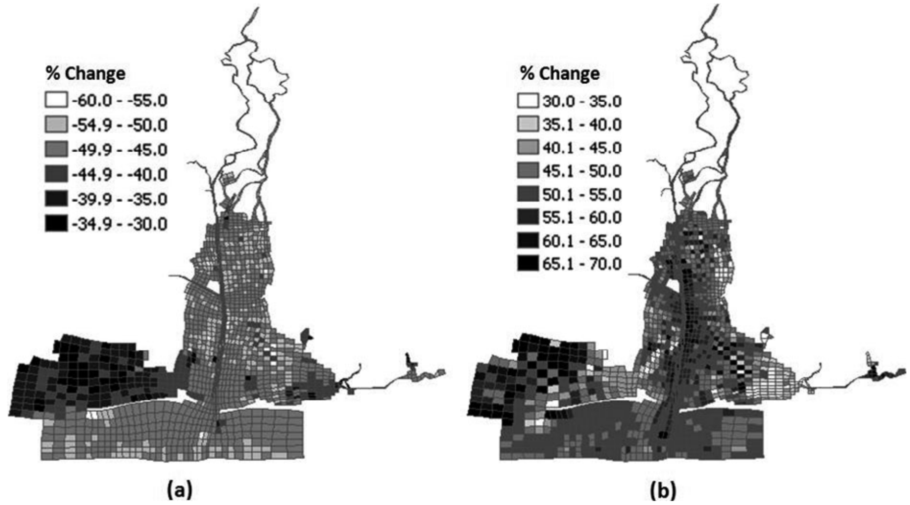

FIGURE 11. Map of the Total Suspended Sediments (TSS) Changes within the Bay Grid Cells for the −50% Sensitivity Analysis Simulation (a) and Map of the TSS Changes within the Bay Grid Cells for the +50% Sensitivity (b).

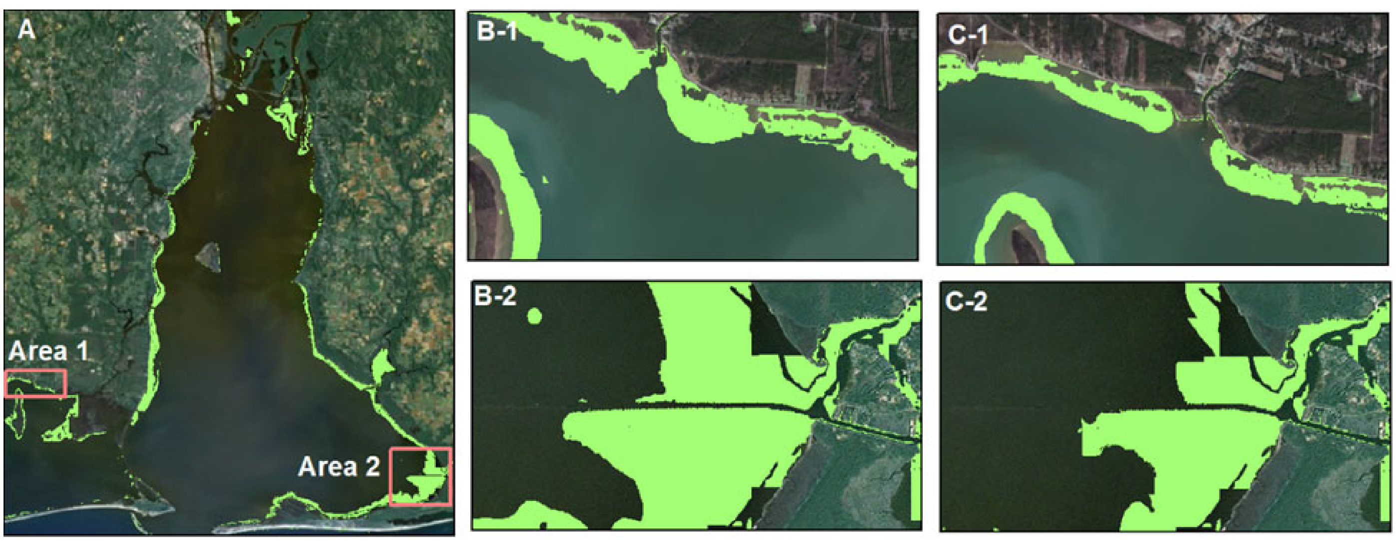

FIGURE 12. Changes to Predicted Submerged Aquatic Vegetation (SAV) Habitat (light) Based on Sensitivity Analysis Scenarios. Baseline conditions shown in (A). Impact of increased suspended sediment can be seen locally at the mouth of the river in Bayou La Batre, Area 1 (B-1 showing an expansion through decreased suspended sediment load scenario and C-1 showing a loss through increased suspended sediment load by 50%). Area 2 shows an increase of habitat area with the decrease of suspended sediments (B-2) resulting in potential for deeper SAV, and a SAV loss with increased suspended sediments (C-2). Other areas of Mobile Bay show little to no change.

RESULTS

Land Cover Land Use

The changes per LCLU category from 1948 to 2001 and 1948 to 2030 for developed and natural LCLU classes show an environment that has become much more urbanized from 1948 to 2001 and the projected effects to 2030 if these trends continue (Figures 13 and 14). LCLU shifted to a more urban environment and freshwater flows into Mobile Bay increased (Figure 15). Since 1948, as Mobile and Baldwin Counties became more urbanized, the urban land covers have replaced forest, agriculture, and pasture land. Increasing urbanization is most prominent in the south and west areas surrounding the city of Mobile, shoreline areas on the Bay's eastern shore, and the beach areas of Gulf Shores and Orange Beach, Alabama. From 1948 to 2001, a 298% increase in urban areas and a decrease of 23% in nonurban areas have occurred (Betancourt *et al.*, 2011).

Fish River, Bayou La Batre, Fowl River, Dog River, and Upper Chickasaw watersheds showed the

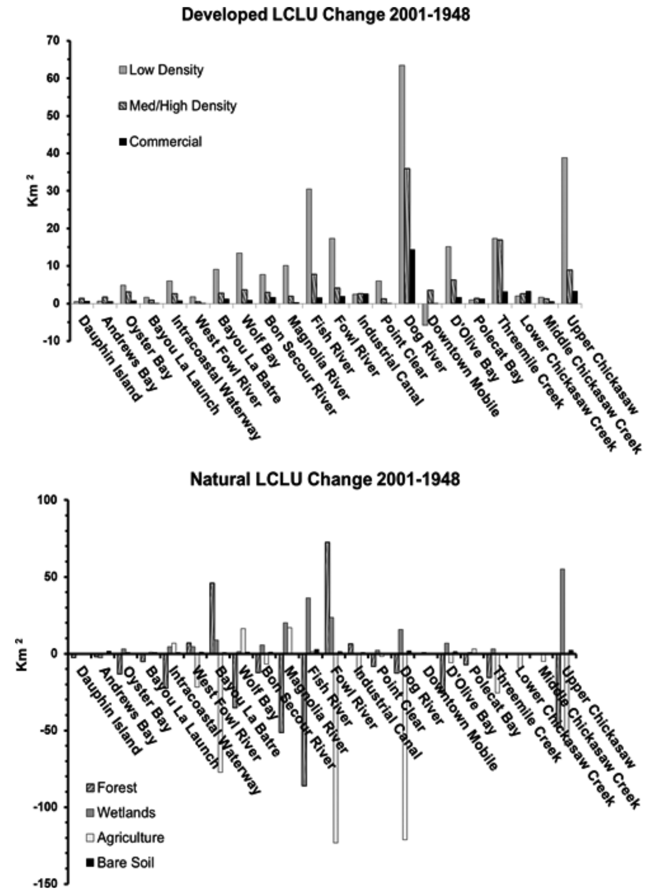

FIGURE 13. Land Cover and Land Use (LCLU) Changes 2001-1948, Developed Classes (top) and Natural Classes (bottom).

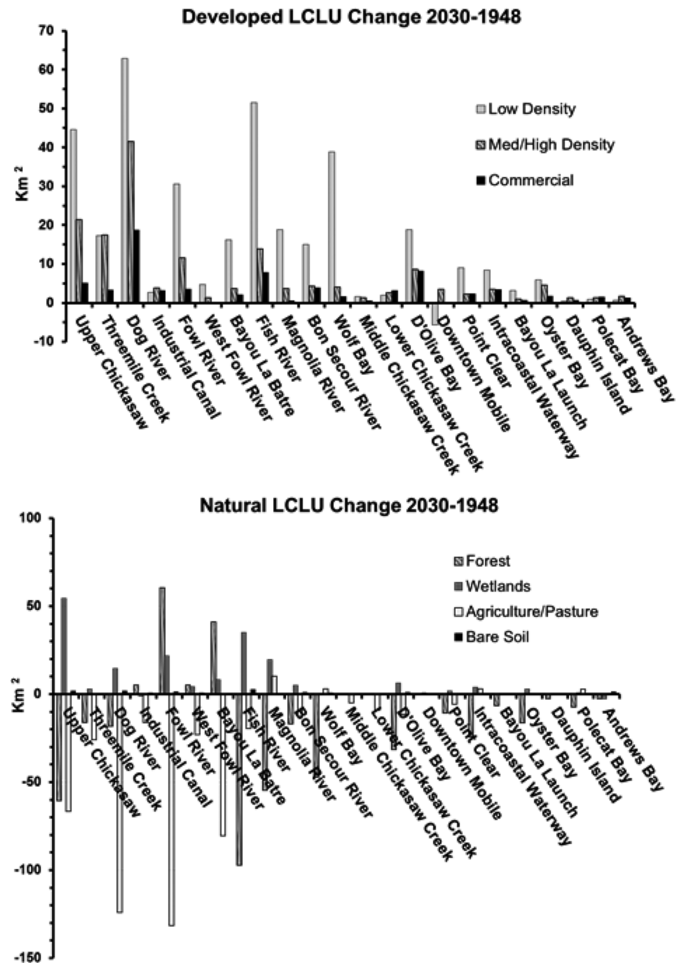

FIGURE 14. Land Cover and Land Use (LCLU) Changes 2030-1948, Developed Classes (top) and Natural Classes (bottom).

largest changes from an agricultural/pasture rural environment due to increasing urbanization. Changes from natural to urban LCLUs increased the velocity of flows and potential erosion of sediments into waterways, whereas decreases in agricultural and pasture land in favor of urbanization or forests reduces the available sediment load. The three major relationships established in the LCLU analysis were that:

1. Increasing urbanization and decreasing agriculture/pasture reduce TSS.
2. Increasing urbanization and decreasing forest raise TSS.
3. Increasing agriculture/pasture and decreasing forest raise TSS.

Streamflow

Modeled streamflow changes using mean annual differences due to the changes in LCLU from 1948 to 2001, and a projected 2030 time frame were evaluated (Figure 15). The largest increases in streamflow are found in the Dog River, Fish River, and Wolf River watersheds. Dog River and Fish River have the highest standard deviation of flows. The highest flow reductions were found in St. Andrews Bay, Fish River, and Upper Chickasaw watersheds. Flow changes from the 1948 baseline to each of the three LCLU scenarios were found to be statistically significant ($\alpha = 0.05$) in all watersheds discharging into the estuary.

The Bayou La Batre and Wolf Bay watersheds streamflow responses were significantly different due to LCLU changes from the 1948 to the 2030 model simulation (Figure 16). Wolf Bay exhibited maximum flows about four times larger than Bayou La Batre and the standard deviation is six times greater. Because the larger watershed size contributes to increases in streamflow, the duration of storm effects was temporally longer for Wolf Bay. The Bayou La Batre watershed reaches periods of decreasing flows after an extended dry period as noted in early May.

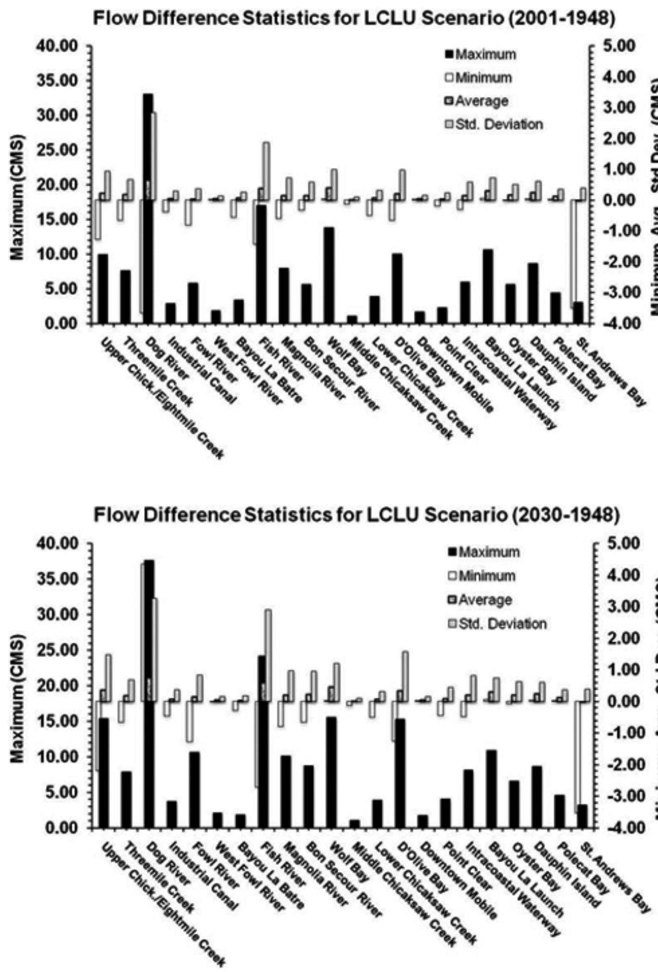

FIGURE 15. Statistics of Daily Streamflow Differences in Land Cover and Land Use (LCLU) Scenarios; 2001-1948 (top) and 2030-1948 (bottom).

Streamflow and TSS Relationship

Sediment loads in metric tons per square kilometer and per day by watershed for the LCLU model simulations were evaluated (Table 5). Results from the 1948 model simulation show that the largest sediment load in tons/day was from the Bayou La Batre watershed. All watersheds in Mobile County and the north shore of Mississippi Sound had decreases in sediment loads from 1948 to 2030. The largest decreases, on the order of about one-third, are found in the Bayou La Batre, West Fowl River, Fowl River, and Upper Chickasaw watersheds. Conversely, watersheds in Baldwin County, Fort Morgan Peninsula, the southeast shore of the Bay, and Dauphin Island show either increases or small changes in sediment loads. Sediment load increases of 50% or larger are found in the Magnolia River, Wolf Bay, Oyster Bay, Dauphin Island, Polecat Bay, and St. Andrews Bay watersheds.

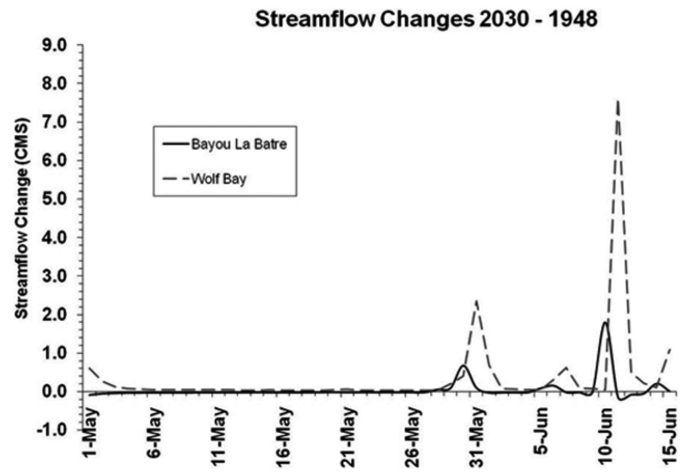

FIGURE 16. Time-Series of Streamflow Differences in Bayou La Batre and Wolf Bay Watersheds.

The watershed model output for streamflow and total TSS concentrations showed significant relationships throughout the estuary. Table 6 indicates the results of linear regression for seven discharge points, which are geographically dispersed throughout the entire estuary. Each of the discharge points, except D'Olive Bay, indicated a R^2 value of 0.45 or higher. The lower correlation is likely the result of the large quantity of wetland area in the D'Olive Bay watershed located in the delta region north of the Bay. Wetlands typically would capture and retain high quantities of the sediment dislodged during rainfall events, and consequently reduce the TSS readings at the discharge points in the estuary. Since most of the wetlands are in the delta region north of Mobile Bay, the most influence of wetlands on TSS is expected to be in this area.

Storm or rainfall intensity significantly influenced TSS concentrations as indicated by the TSS fluctuations for rainfall events of small, moderate, and high intensity for each LCLU simulation in the Wolf Bay and Bayou La Batre watersheds (Figure 17). For this evaluation, small intensity is a precipitation event of 19.05 mm (0.75 inches) over two days, a moderate event is 64.01 mm (2.52 inches) over four days, and a high-intensity event is 114.3 mm (4.5 inches) over half a day. Small-to-moderate rainfall events show a similar pattern in each watershed with the TSS concentrations increasing from 1948 to 2001 to 2030. However, for the high-intensity rainfall event in Bayou La Batre, the TSS levels are much higher in the 1948 LCLU simulation than the other years, likely due to the much higher acreage in agriculture and pasture land use at this time. In Wolf Bay, the TSS concentrations are very similar for each LCLU simulation possibly indicating competing influences between changing acreages of

TABLE 5. Sediment Loads by Watershed and Land Cover and Land Use Year.

	Metric Tons/km^2				Metric Tons/Day			
Watershed	1948	1992	2001	2030	1948	1992	2001	2030
Upper Chickasaw	2,342.5	750.0	728.0	713.6	10.7	3.4	3.3	3.3
Threemile Creek	1,769.8	1,296.4	1,317.5	1,314.3	7.6	5.7	5.8	5.8
Dog River	2,822.8	1,345.5	1,237.0	1,303.0	32.7	16.5	15.2	16.0
Industrial Canal	3,420.0	1,421.5	1,208.8	1,283.4	40.9	17.2	14.6	15.5
Fowl River	2,943.4	1,112.5	863.2	1,100.8	50.9	19.2	14.9	19.0
West Fowl River	1,265.4	542.3	372.0	365.8	56.6	24.8	17.0	16.7
Bayou La Batre	2,245.6	906.3	812.2	813.8	136.2	56.6	50.7	50.8
Fish River	1,258.7	1,356.4	1,317.1	1,109.7	32.3	34.9	33.9	28.6
Magnolia River	1,635.5	2,738.2	2,683.7	2,624.9	97.1	162.8	159.6	156.1
Bon Secour	2,083.1	2,340.4	2,392.4	2,427.3	74.7	86.9	88.9	90.2
Wolf Bay	788.4	1,211.9	1,182.0	1,096.9	33.1	60.2	58.4	54.2
Middle Chickasaw	2,025.6	522.2	524.1	501.2	7.6	2.0	2.0	1.9
Lower Chickasaw	2,556.2	913.8	885.5	877.7	15.9	5.7	5.5	5.5
D'Olive Bay	2,133.4	2,160.8	2,186.1	2,107.9	106.8	110.8	112.1	108.1
Downtown Mobile	1,311.7	1,045.9	1,058.2	1,060.2	19.6	16.5	16.7	16.7
Point Clear	1,846.9	2,101.4	2,112.8	1,932.8	48.7	55.4	55.7	51.0
Intracoastal Waterway	933.8	2,052.0	2,015.7	1,927.2	31.7	73.2	71.9	68.8
Bayou La Launch	557.8	595.9	549.4	551.6	7.4	12.7	11.7	11.7
Oyster Bay	570.0	1,004.7	927.2	981.4	18.2	35.0	32.3	34.2
Dauphin Island	476.1	391.7	414.8	389.1	1.9	4.0	4.3	4.0
Polecat Bay	628.9	696.9	1,267.8	1,244.8	6.2	8.8	16.0	15.7
St Andrews Bay	359.0	672.8	624.0	623.7	5.0	9.7	9.0	9.0

TABLE 6. Watershed Model Statistical Relationships for Streamflow and Total Suspended Sediments.

	R^2			
Watershed	1948	1992	2001	2030
Bayou La Batre	0.47	0.50	0.49	0.48
Bon Secour	0.60	0.63	0.63	0.64
Dog River	0.66	0.59	0.56	0.52
D'Olive Bay	0.14	0.13	0.13	0.12
Fowl River	0.67	0.68	0.68	0.62
Magnolia River	0.45	0.47	0.47	0.47
Wolf Bay	0.55	0.54	0.54	0.53

Note: All values are significant at a p-value less than 0.001.

agriculture/pasture and developed land among the LCLU scenarios.

Hydrodynamic Effects on TSS

Changing from natural to urban LCLU increases the velocity of flows and potential erosion of sediments into waterways. Decreasing the agricultural and pasture land in favor of urbanization or forests reduces the available sediment load, which would directly affect the TSS within the same column of water assuming no other factors have changed. For example, the Bayou La Batre subwatershed has been significantly losing agricultural land over the last 60 years and is expected to continue over the next decades (Figures 13 and 14). Also, the results of the hydrodynamic model showed that the average water column TSS decreased the most in the grid cell closest to the Bayou La Batre subwatershed's discharge point into Mobile Bay (Figure 18). On the other hand, the Wolf Bay subwatershed has been gaining agricultural land at the expense of forest land over the last 60 years, which could explain why the average water column TSS increased the most in the grid cell closest to Wolf Bay subwatershed's discharge point.

The dominating factors of circulation and water quality formation of Mobile Bay are bathymetry, freshwater flow (streamflow), tidal surface elevation, wind, solar radiation, air temperature, and sediments (TetraTech, 2006). None of these factors, except freshwater flow and sediments, have changed between the different LCLU model simulations. Thus, to better understand the effect of hydrodynamics on TSS concentrations, we performed linear regression on the watershed model streamflow output *vs.* the hydrodynamic TSS output for eight geographically dispersed discharge points around the Bay (Wolf Bay, Bon Secour River, Magnolia River, D'Olive Bay, Bayou La Batre, Industrial Canal, Dog River, and Upper Chickasaw). The results showed statistically significant relationships (p = 0.05) and moderate to good correlations (r = 0.33-0.62) between streamflow and TSS at all the grid cells/discharge points except those close to the mouth of the Bay (i.e., near the GOM), where the tidal currents dominate circulation. The tidal

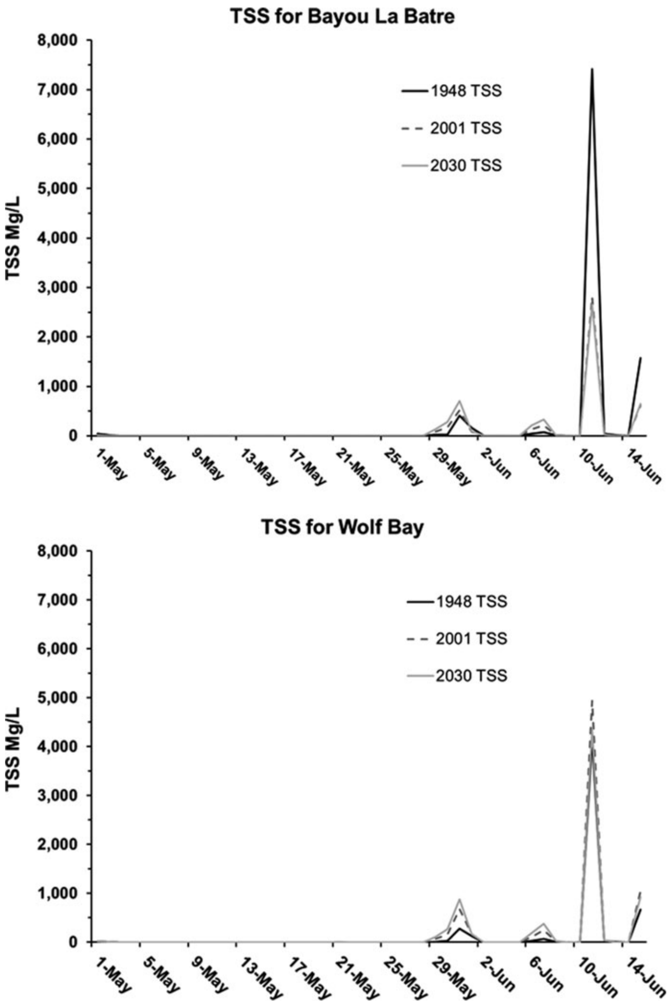

FIGURE 17. Total Suspended Sediments (TSS) Concentrations per Land Cover and Land Use Simulation at Wolf Bay and Bayou La Batre Watershed Discharge Points.

currents are responsible for deepwater intrusions from the GOM, which cause higher circulations and decrease the TSS concentrations in individual grid cells (Schroeder and Wiseman, 1986; TetraTech, 2006). That was even more evident when we compared the results among the grid cells near the mouths of individual Bays (at Bayou La Batre, Magnolia, and Wolf Bay). The results showed that the relationship between streamflow and TSS was higher in Wolf Bay and Bon Secour River than it was in Bayou La Batre where tidal currents have the most effect among the three sites.

Salinity and Temperature

Changes in the water column for temperature due to LCLU-driven changes in streamflow were very small (i.e., $\leq$0.5°C on average). For salinity changes in the water column, 17 of the 22 discharge locations

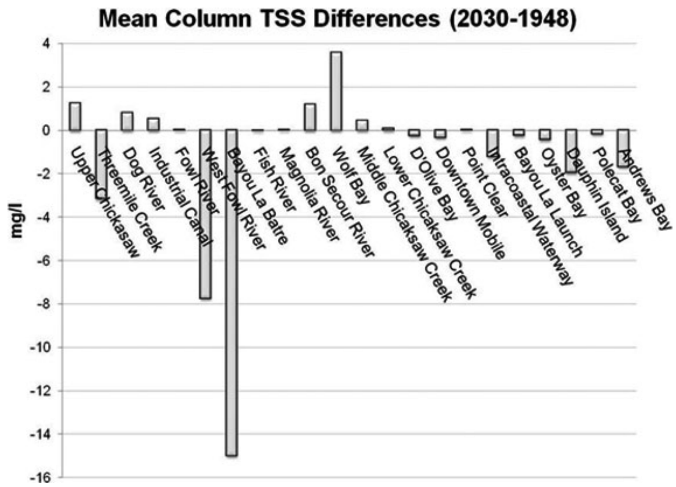

FIGURE 18. Mean Total Suspended Sediments (TSS) Differences per Discharge Point Comparing the 1948 and 2030 Land Cover and Land Use Simulations for May.

had changes of less than 0.5 PSU and the other 5 locations had changes of less than 7 PSU on average. However, the water column changes for both temperature and salinity were well within the physiological tolerances of the SAV species studied (e.g., Fonseca *et al.*, 1998). Therefore, analysis on these variables was not performed.

Habitat Suitability Analysis

From 1948 to 2030, few areas were predicted to change in habitat suitability based on changes in the light regime. However, changes were more evident in coastal areas adjoining the Bay (Figure 19). The largest changes in extent can be seen in the area of Bayou La Batre, which has a predicted expansion of area due to an increase of areas meeting the 19% SI threshold.

Between 1948 and 2030, two sites, Bayou La Batre and Wolf Bay, both external to Mobile Bay proper, exhibited predicted changes in maximum SAV depth greater than 0.1 m (Figure 20). In Bayou La Batre, the maximum depth at which SAV grows was predicted to increase from 1948 to 2030, potentially expanding the total area of SAV habitat. In Wolf Bay, based on light attenuation, the maximum depth at which SAV grows was predicted to decrease from 1948 to 2030, potentially decreasing SAV habitat.

DISCUSSION AND CONCLUSIONS

The modeling system and subsequent results provide new knowledge for the Mobile Bay estuary and

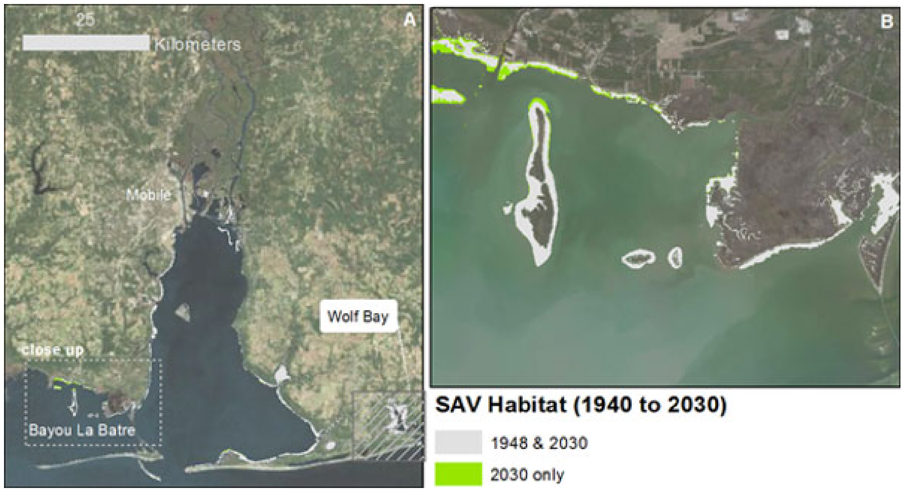

FIGURE 19. Submerged Aquatic Vegetation (SAV) Habitat Change from 1940 to 2030. While most of Mobile Bay (A) shows little change, areas near Bayou La Batre (B) show potential expansion of habitat. A bathymetric dataset of sufficient precision to map measured changes was not available for Wolf Bay (shaded area in A), although our rapid assessment indicated potential habitat decline.

the linkages between LCLU and SAV habitat suitability and the modeling system is a potential tool to enhance conservation decision making. However, it is imperative to effectively use this modeling system and/or the results to understand that the models and subsequent results are conditional on the inherent assumptions within the modeling system and limited by uncertainty or range of error. The sensitivity analysis that found variances in habitat suitability of 5-10% should be considered when interpreting modeling results from this study and future modeling scenarios (Crosetto *et al.*, 2000).

Modeled results indicate that LCLU changes are increasing freshwater flows into Mobile Bay that are altering temperature, salinity, and TSS. However, these changes cannot solely account for the changes seen in SAV extent in most areas in Mobile Bay. Discharge from the Tensaw-Mobile River confluence at the head of the Bay remains the largest source of material influx, with local watersheds and rivers contributing a relatively minor load (Schroeder *et al.*, 1990). In our study, changes in LCLU within the multiple smaller watersheds surrounding Mobile Bay resulted in very minor changes to SAV habitat within the Bay itself. It is likely that the “signal” from LCLU change was overwhelmed in much of the Bay by the load from the larger river discharge. This suggests to us that SAV distribution in much of the Bay is influenced by this major discharge, and the incremental changes we modeled have little effect. However, both Bayou La Batre and Wolf Bay, lying east and west of Mobile Bay, respectively, did exhibit changes in potential maximum water depth and SAV habitat extent. Tidal currents likely contributed to the increasing depth for suitable habitat at Bayou La Batre through increased mixing and dissipation of TSS, while Wolf Bay was more isolated from tidal effects due to its location north of the Intracoastal Waterway and surrounding shoreline configuration.

Including dynamics from Mobile River in the assessment would likely more accurately represent the system, although it would have made it more difficult to isolate the response to local watershed LCLU changes. Also, since the 1948 LCLU coverage did not include a wetlands class, the coverage was normalized to the 1992 wetlands extent. This likely underestimates the contribution of wetlands in 1948 to reduce streamflow and affect suspended sediment loads.

Many urban estuaries, such as Tampa Bay, Chesapeake Bay, and Puget Sound, are utilizing SAV extent as an indicator of aquatic ecosystem health relative to anthropogenic changes in the watersheds (Johansson and Ries, 1996; Orth *et al.*, 2006; Puget Sound Partnership, 2012). In our study, observed and predicted changes in LCLU had a relatively minor predicted impact on SAV extent. The influence of local watersheds on adjacent SAV extent is likely a function of specific site location within the Bay and the magnitude of LCLU change. For example, the most significant changes in maximum depth of habitat for SAV occurred in areas outside Mobile Bay proper. While areas of high light do not mean that SAV should necessarily exist at that location, as there are other nearshore habitats (mudflats, oyster) in these zones, we expect that stable SAV meadows should occur in and around areas with high light availability. Reduction of high light areas around existing beds should be a concern for future maintenance of the habitat.

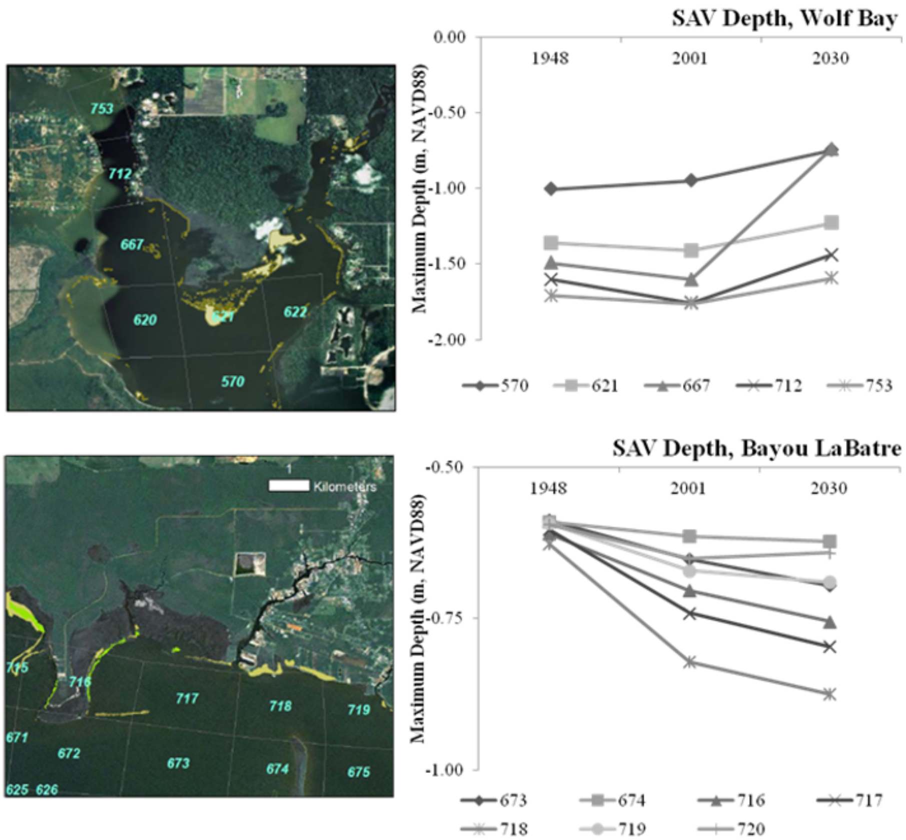

FIGURE 20. Wolf Bay (upper left) and Bayou La Batre (lower left) with the Labeled ID of Each Cell Shown. Present day (2008-2009) submerged aquatic vegetation (SAV) classification in bright green overlain on 2001 projection of areas with sufficient light (light brown) (see color figure in online version). These are areas where the depth limit of SAV either would recede or expand by more than 0.1 m vertical. Projections (upper and lower right) show the projected maximum depth of SAV for each cell through the time series.

Our initial examination of field-collected water properties indicates that TSS is the largest contributor to turbidity in Mobile Bay. The secondary component to light attenuation beyond TSS in many estuaries is nutrient-driven algae blooms. Based on the field measurements collected between November 2007 and July 2008, turbidity appears to be dominated by TSS in Mobile Bay, although DOC and occasional algal blooms contributed to light attenuation and increased turbidity. The National Estuarine Eutrophication Assessment (Bricker *et al.*, 1999) indicated a low degree of loss of SAV attributed to eutrophication. The importance of nutrient loading may change with LCLU change, especially if changes are correlated with an increase in nutrient rich runoff.

Our model results appear to indicate that local management and development regulations designed to protect nearshore environments may be most effective for small inlets that are under less influence of the Mobile Bay watershed, such as the Wolf Bay and Bayou La Batre areas that showed the highest sensitivity to water depth changes driven by LCLU change. This is a localized adaptation of a principle in resource management for many species that underline the critical need of multiscale plans and management (Fausch *et al.*, 2002; Orth *et al.*, 2006), as indeed the influence of a process like sediment transfer and deposition is not homogeneous across a landscape, rather its impacts are determined by a site's location in the landscape relative to other factors and drivers (e.g., Turner, 1989).

For other estuarine systems, our model results underline the importance of scale and magnitude. Small-scale restoration and conservation projects may not have a significant impact on SAV health when acting on relatively small watersheds within larger systems. Importance of managing at the appropriate scale is echoed by Orth *et al.* (2006) who point to the management need and challenge of implementing plans for sediment and nutrient reduction across jurisdictional boundaries. However, one positive note is that improvements in smaller watersheds, located within a major watershed, could be potentially very successful.

Maintaining and increasing the areal extent of forests are potentially the most effective LCLU change strategy to improve water clarity in shallow aquatic ecosystems, but these changes would need to occur throughout the watershed, not just for local watersheds. Among the factors of salinity, temperature and TSS that we evaluated, TSS showed the greatest response to LCLU changes. The more urbanized watersheds of Dog River, Upper Chickasaw, and Downtown Mobile showed above average increases in streamflow discharge and fluxes from 1948 to 2001. Previous research has confirmed that increased percentages of urban development are associated with higher runoff volumes and that a higher percentage of agriculture and pasture LCLU is correlated with higher sediment loads (Thom *et al.*, 2001; Betancourt *et al.*, 2011). Watersheds experiencing shifts to more urban land such as Fish River and Fowl River were also among the largest increases in streamflow discharges. While correlations between streamflow increases and higher TSS concentrations were found throughout the estuary, the magnitude and temporal characteristics of the streamflow and the LCLU characteristics affected the relationship. Also, wetlands in watersheds north of the Bay and tidal circulation in the extreme southern extent of the estuary tended to reduce TSS levels and subsequent correlation with streamflow discharges.

With increasing urbanization in watersheds surrounding Mobile Bay, development of this modeling system will provide resource managers with useful information to understand and manage shallow water habitat types that provide services critical to Bay and GOM ecosystems and communities. The Mobile Bay National Estuary Program and the Alabama Department of Conservation and Natural Resources have concentrated resources and efforts to determine trends related to SAV distribution in Alabama's coastal waters. The results of this study provide an additional data component for resource managers to consider when making decisions about SAV restoration and protection.

This modeling system can be used to evaluate various environmental scenarios and provide data for analysis to determine cause/effect relationships of changes in distribution/occurrence of SAV. There are a variety of modeling approaches for species habitat. Suggestions for model preference in landscape ecology tend toward simpler models if there is greater uncertainty in the system (Peters *et al.*, 2004). Using a threshold approach for ecological assessment, our modeling system provides a rapid, straightforward manner to map spatial responses of various SAV species, although it does not provide a ranking or continuum of habitat quality. Coupling this analysis with a mechanistic model for plant growth and survival would provide a more complete and responsive link to the water property changes in the system and may be preferable — if information and understanding at appropriate scales were available.

In addition to LCLU change, climate change is a major potential influence on Mobile Bay and its adjacent estuaries that needs further study (Burkett and Davidson, 2012). IPCC projections to 2050 are for a warmer and drier climate for the northern Gulf coast (Van Vliet *et al.*, 2013). According to the IPCC fourth Assessment Report (IPCC, 2007), mean sea level is projected to rise from 1990 to 2095 by 0.23-0.51 m for the A2 business as usual scenario, by 0.21-0.48 m for the A1B weak stabilization scenario, and by 0.18-0.38 m for the B1 strong stabilization scenario. Preliminary research indicates that salinity is more sensitive to climate changes than temperature or TSS. Further evaluation of climate effects on the estuarine ecosystems could be beneficial in the development of climate adaption plans.

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